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## Soil health recovery after grassland reestablishment on cropland: The effects of time and topographic position

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# Soil health recovery after grassland reestablishment on cropland: The effects of time and topographic position

## Abstract

The Conservation Reserve Program (CRP) is a U.S. federal land conservation program that incentivizes grassland reestablishment on marginal lands. Although this program has many environmental benefits, two critical questions remain: does reestablishing grasslands via CRP also result in soil health recovery, and what parts of restored fields (i.e., topographic positions) recover the fastest? We hypothesized that soil health will recover over time after converting cropland to CRP grassland and that recovery will be greatest at higher topographic positions. To test this, we sampled 241 midwestern U.S. soils along a grassland chronosequence (0–40 yr, including native grasslands) and at four topographic positions (i.e., a chronotoposequence). Soils were measured for bulk density, maximum water holding capacity (MWHC), soil organic C (SOC), extractable inorganic N, potentially mineralizable C (PMC), and N. Native grasslands had superior soil health compared with cropland and most CRP soils, and even 40 yr since grassland reestablishment was not adequate for full soil health recovery. Topographic position strongly influenced soil health indicators and often masked any CRP effect, especially with MWHC and SOC. However, PMC (a measure of active C) responded most rapidly to CRP and consistently across the landscape and was 26–34% greater 19–40 yr after grassland reestablishment. Reestablishing grasslands through CRP can improve soil health, although topographic position regulates the recovery, with greatest improvements at shoulder slope positions. Patience is needed to observe changes in soil health, even in response to a drastic management change like conversion of cropland to CRP grassland.

## Disciplines

Agriculture | Agronomy and Crop Sciences | Ecology and Evolutionary Biology | Soil Science

## Comments

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# Soil health recovery after grassland reestablishment on cropland: The effects of time and topographic position

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## Abstract

The Conservation Reserve Program (CRP) is a U.S. federal land conservation program that incentivizes grassland reestablishment on marginal lands. Although this program has many environmental benefits, two critical questions remain: does reestablishing grasslands via CRP also result in soil health recovery, and what parts of restored fields (i.e., topographic positions) recover the fastest? We hypothesized that soil health will recover over time after converting cropland to CRP grassland and that recovery will be greatest at higher topographic positions. To test this, we sampled 241 midwestern U.S. soils along a grassland chronosequence (0–40 yr, including native grasslands) and at four topographic positions (i.e., a chronotoposequence). Soils were measured for bulk density, maximum water holding capacity (MWHC), soil organic C (SOC), extractable inorganic N, potentially mineralizable C (PMC), and N. Native grasslands had superior soil health compared with cropland and most CRP soils, and even 40 yr since grassland reestablishment was not adequate for full soil health recovery. Topographic position strongly influenced soil health indicators and often masked any CRP effect, especially with MWHC and SOC. However, PMC (a measure of active C) responded most rapidly to CRP and consistently across the landscape and was 26–34% greater 19–40 yr after grassland reestablishment. Reestablishing grasslands through CRP can improve soil health, although topographic position regulates the recovery, with greatest improvements at shoulder slope positions. Patience is needed to observe changes in soil health, even in response to a drastic management change like conversion of cropland to CRP grassland.

## 1 | INTRODUCTION

Grassland reestablishment under the Conservation Reserve Program (CRP) is the largest U.S. land retirement pro-

**Abbreviations:** BD, bulk density; CRP, Conservation Reserve Program; GPS, global positioning satellite; MBC, microbial biomass carbon; MWHC, maximum water holding capacity; PMC, potentially mineralizable carbon; PMN, potentially mineralizable nitrogen; SHRS, Soil Health Recovery Score; SOC, soil organic carbon.

gram, initiated in the 1985 Farm Bill to convert highly eroded cropland to a permanent grassland cover (Morefield, LeDuc, Clark, & Iovanna, 2016; Skold, 1989). The CRP provides many ecological benefits such as preservation and restoration of forests and wetlands, wildlife habitat, improved air and water quality, and carbon (C) sequestration (Allen, & Vandever, 2012; FAPRI, 2007; Follett, Pruessner, Samson-Liebig, Kimble, & Waltman, 2001; Li et al., 2017a). Despite these benefits, the amount of CRP

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land in the United States has steadily declined from its peak enrollment of 14.9 million ha in 2007 to 9.5 million ha in 2017 (USDA-FSA, 2017). Declines in CRP enrollment are likely to affect those well-established ecosystem benefits, but the impacts on soil health remain unclear for those lands being converted out of CRP and for conversion of existing cropland to new CRP grasslands.

Globally, ~70% of native grasslands have been either converted to cropland or to other land uses (Ramankutty, Evan, Monfreda, & Foley, 2008). Nearly all native grasslands have been converted to agriculture in the U.S. Midwest Corn Belt, with only 1% of native grasslands remaining today (Samson & Knopf, 1994; Wright & Wimberly, 2013). Compared with native grasslands, long-term cultivation has reduced soil organic matter (SOM) between 30 and 60% (Kucharik et al., 2001; Potter, Torbert, Johnson, & Tischler, 1999; Schlesinger, 1986), altered soil microbial communities (Allison, Miller, Jastrow, Matamala, & Zak, 2005), accelerated cycling and loss of nutrients (Burke, Lauenroth, & Coffin, 1995a), reduced nutrient retention (Kemp & Dodds, 2001), and compacted soil (Murphy, Foster, Ramspott, & Price, 2004). The exact reasons for loss in SOM, along with concomitant declines in soil health, are unknown. Likely reasons include disturbance due to aggregate disruption through tillage (Blanco-Canqui & Lal, 2009; Gebhart, Johnson, Mayeux, & Polley, 1994), artificial drainage (Arabi, Stillman, & Govindaraju, 2006; Du, Arnold, Saleh, & Jaynes, 2005), decreased plant biomass inputs (Richter, Babbar, Huston, & Jaeger, 1990; Rosenzweig, Carson, Baer, & Blair, 2016), and perhaps even liming (Chan & Heenan, 1999; Wang, Tang, Baldock, Butterly, & Gazey, 2016). Returning cropland to perennial grassland cover under CRP has been proposed as one strategy for regenerating SOM lost due to cultivation (Gebhart et al., 1994; Karlen et al., 1999; Reeder, Schuman, & Bowman, 1998). This is critical for two reasons: (i) SOM plays a central role in soil health (Karlen et al., 1999; Post & Kwon, 2000) in that it is directly linked to many soil ecosystem services, such as nutrient storage and supply, water storage and regulation, and habitat for organisms (FAO and ITPS, 2015; Lal, 2015, 2018), and (ii) the conversion of cropland to perennial grasslands has been suggested as a strategy to mitigate global climate change via soil C sequestration (Paustian et al., 2016). Although it is not economically feasible to restore large swaths of cropland to grassland (e.g., CRP), subfield profitability studies indicate that converting small, unprofitable portions of fields in the U.S. Midwest may provide substantial environmental benefits with little impact on farm profitability (Brandes et al., 2018).

Previous studies have taken a chronosequence approach in evaluating the CRP effect on soil health or have compared native grasslands with cropped soils (Baer, Kitchen, Blair, & Rice, 2002; Knops & Tillman, 2000; Matamala, Jas-

### Core Ideas

- We evaluated soil health recovery after converting cropland to grassland.
- Time since reestablishment and topographic position regulated soil health recovery.
- After 40 yr of grassland establishment, no soil health measure approached native grassland.
- Active carbon responded most rapidly and consistently, across topographic positions.
- Shoulder slope position showed greater improvements in soil health with time.

trow, Miller, & Garten, 2008; Mensah, Schoenau, & Malhi, 2003; Reeder et al., 1998; Rosenzweig et al., 2016). However, most of these studies have focused on insensitive or slower-changing soil health indicators (e.g., total soil organic C and N pools) and have reported inconsistent findings. For example, some studies found that CRP can increase soil organic C (SOC) by 20–110 g m<sup>-2</sup> yr<sup>-1</sup> (Gebhart et al., 1994; Knops & Tillman, 2000), whereas other studies found no change after CRP establishment (Breuer, Huisman, Keller, & Frede, 2006; Burke et al., 1995a, 1995b) or even decreases up to m<sup>-2</sup> yr<sup>-1</sup> in CRP compared with cropland (McKee, Brye, & Wood, 2019). A handful of studies have evaluated more management-sensitive active C pools, such as potentially mineralizable C (PMC) and microbial biomass C (MBC) (Baer et al., 2002; Karlen et al., 1999; Matamala et al., 2008; Rosenzweig et al., 2016; Scott, Baer, & Blair, 2017). These studies have found nearly double PMC concentrations and ~50% greater MBC in soils under CRP compared with cropland (Baer et al., 2002, 2010; Karlen et al., 1999), suggesting that these active C pools can be more sensitive as indicators of soil health than total SOC. Inconsistent findings with total SOC are likely due to a combination of large soil heterogeneity within a field, error associated with measurements of SOC, and lack of sensitivity to management changes (Goidts, Van Wesemael, & Crucifix, 2009; Stewart, Paustian, Conant, Plante, & Six, 2007; Yu, Lu, Cao, & Tian, 2018).

Topographic position, one of five soil-forming factors (Jenny, 1941), is a well-known driver of soil variation at the field scale (De, Saha, & Chakraborty, 2014; Tsui, Chena, & Hsieh, 2004; Wiesmeiera et al., 2019; Zilverberga et al., 2018). Indeed, large variance in soil properties across topographic positions can often mask any management effect if not taken into account (Kravchenko, Robertson, Thelen, & Harwood, 2005, 2006; Quigley, Rivers, & Kravchenko, 2018). Thus, although CRP grassland establishment may improve soil health, the strong influence of topographic position may confound interpretations and render land management

recommendations ineffective or difficult, especially in fields with large topographic variation.

Given the importance of both years-since-reestablishment and topographic position on the effect of CRP on soil health, we designed a “chronotoposequence” study in the Midwestern U.S. Corn Belt. To our knowledge, we are the first study to use this chronotoposequence approach in reestablished CRP grasslands, where we analyze the effect of years-since-reestablishment and interaction with four topographic positions. Our overall objective was to quantify changes in physical (e.g., maximum water holding capacity [MWHC] and bulk density [BD]), chemical (e.g., SOC and extractable inorganic N), and biological (e.g., PMC and potentially mineralizable N [PMN]) soil health indicators with grassland reestablishment via CRP. Across the chronosequence, we used cropland as a baseline and native grassland as a soil health benchmark or goal. More specifically, our research questions were: (i) Can reestablishing CRP grasslands also restore soil health, and over what time scales? and (ii) What role does topographic position have on regulating effects of CRP on soil health? Based on previously published literature, we hypothesized that MWHC, SOC, PMC, and PMN under reestablished CRP perennial grasslands will increase with time, but BD and extractable inorganic N (ammonium plus nitrate [ $\text{NH}_4^+ - \text{N} + \text{NO}_3^- - \text{N}$ ]) will decrease with CRP age. Due to transport of fine SOM-associated soil particles down slope by erosion as well as other contributing mechanisms (e.g., increased productivity and decreased decomposition), accumulation of SOM often occurs at lower topographic positions under cultivation (De et al., 2014; Olson, Al-Kaisi, Lal, & Cihacek, 2016a, 2016b; Tang, Liu, Liu, & Zhou, 2010). With less SOM at higher topographic positions, we hypothesized that changes in SOM after CRP would be easier to detect because they would have greater room for improvement or potential for increased C in physico-chemically protected fractions (Castellano, Mueller, Olk, Sawyer, & Six, 2015; Stewart et al., 2007; Zilverberga et al., 2018). Conversely, we expected that lower topographic positions, characterized by having greater SOM than higher positions, would show less change in soil health following grassland restoration.

## 2 | MATERIALS AND METHODS

### 2.1 | Study site description

This study was part of a larger regional study to establish baseline data and sampling sites for potential future monitoring of soil C change of land parcels (Riipel, 2009). Nineteen study sites were included from the north-central Iowa counties of Clay, Emmet, Kossuth, and Palo Alto and southern Minnesota counties of Jackson and Cottonwood, which

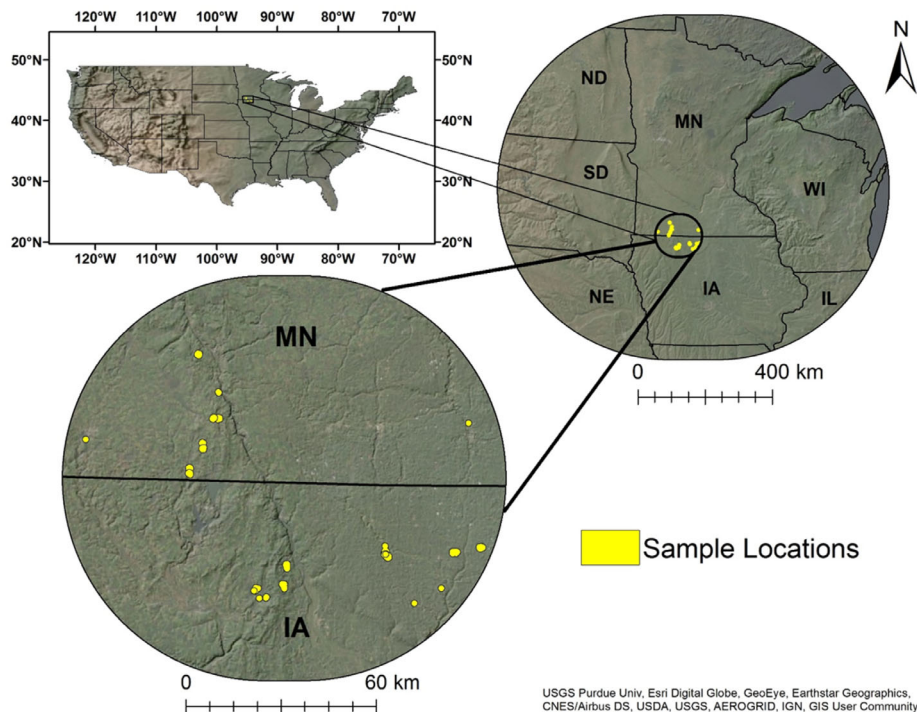
fall within the Prairie Pothole Region in the northern Great Plains of the United States (Figure 1). Of these sites, three were cultivated, 13 were reestablished grasslands enrolled in CRP, and three were native grasslands. A total of 241 separate soil cores were sampled across the 19 sites (Figure 1). All sites were within the Major Land Resource Area Map Unit 103 (USDA, 1981), with an average annual rainfall of 625–850 mm, average annual temperature of 6–9 °C, and average frost-free period of 130–160 d (Riipel, 2009; USDA, 1981). Soils at the sites were from a relatively narrow range of soil texture classes. Nearly 60% of our soil textures were in loam class, and 75% of our samples had clay between 17 and 34% (Supplemental Figure S1).

The composition of the vegetation varied within each field. The three most dominant species in the reestablished CRP grasslands were bromegrass (*Bromus inermis* L.), Canada wild rye (*Elymus Canadensis* L.), and big bluestem (*Andropogon gerardii* L.). The three most dominant species in the native grasslands were Kentucky bluegrass (*Poa pratensis* L.), bromegrass (*Bromus inermis* L.), and Timothy grass (*Phleum pratense* L.). More details of the study sites, including soil map unit symbols, soil types, and slopes, can be found in Supplemental Table S1 and in Riipel (2009).

The study was designed as a “chronotoposequence” based on time since CRP conversion and four different topographic positions. The chronological treatments included a cropland treatment, multiple years of restoration, and a native grassland as a soil health benchmark or goal. The years since reestablishment and number of soil samples collected include: 0 (croplands,  $n = 42$ ), 2 ( $n = 12$ ), 3 ( $n = 28$ ), 6 ( $n = 14$ ), 10 ( $n = 33$ ), 13 ( $n = 23$ ), 19 ( $n = 36$ ), 25 ( $n = 15$ ), and 40 yr ( $n = 14$ ) in CRP and native grassland ( $n = 24$ ). The CRP grassland ages were determined from records provided by the landowners and land managers. Lands that were enrolled for over 25 yr were established as U.S. Department of Natural Resources or the U.S. Fish and Wildlife Service Management Areas prior to the enactment of CRP in 1985. These areas were taken out of crop production and restored as native prairie grasslands for waterfowl and pheasant habitat. Because they had a similar management to CRP lands, they were included in the analysis to capture effects of grassland restoration exceeding 25 yr.

The fields selected ranged in area from 24.3 to 64.8 ha. Total sampling area was 805.3 ha, and each sampling point was located for every 4.05 ha of a field. The sampling points were selected on a representative basis from the soil types and topographic positions present in the field where the soil types are generally related to topographic positions. The slopes of the toposequences ranged from 0 to 25% (Supplemental Table S1; Riipel, 2009). The topographic positions were shoulder- ( $n = 43$ ), back- ( $n = 47$ ), foot- ( $n = 34$ ), and toe-slopes ( $n = 17$ ).





**FIGURE 1** Map showing the sampling locations in southern Minnesota (MN) and northern Iowa (IA). Map was produced by Meyer P. Bohn at Iowa State University, IA, USA.

## 2.2 | Soil sampling, processing, and basic analyses

Soils were sampled between July and August of 2008. The location of each soil sample was logged with personal digital assistants (Dell Axim X50) equipped with a global positioning satellite (GPS) Farm Works GPS receiver (model number D157N) or the GlobalSat GPS Compact Flash (model number BC-337) marked with a flag to denote the GPS location. A hand probe (1.9 cm diameter, 30.5 cm length) was used to collect five subsamples (0–15 cm) throughout all the field-sampling locations by taking the first sample directly to the north of a flag. The next four samples were sampled in a clockwise circle around the flag at equal distances (~5 m) and combined to form a composite sample (Cihacek, Botnen, & Steadman, 2010). Composited soil sample bags for each location were transported in a cooler to the laboratory for refrigeration at 4 °C until analyses.

The sample bags (previously tared) were weighed for total soil mass for BD determination using a core method (Blake & Hartge, 1986). Subsamples were taken from each composited sample bag to determine gravimetric soil moisture content as described by Gardner (1986) and used to correct the soil mass to determine BD. After soil moisture had been measured, the remaining soil was air dried and sieved (<2 mm). Total C content was measured using high-temperature combustion in a Skalar Primacs C analyzer (Skalar Inc.) as described by Nelson and Sommers (1996) and Cihacek and Jacobson (2007).

Inorganic C was determined by phosphoric acid addition to dissolve soil carbonates, and the same instrument used for total C detection measured the CO<sub>2</sub> evolved. Total SOC concentration was calculated as the difference between total C and inorganic C. Further, SOC stock was calculated as the product of SOC concentration, BD, and soil depth (De et al., 2014).

## 2.3 | Potentially mineralizable carbon and nitrogen incubation

Soils were analyzed for PMC and PMN using a 107-d aerobic incubation (similar to Paul, Harris, Collins, Schulthess, & Robertson [1999] and McDaniel & Grandy [2016]) in 2017 (10 yr later). Although air-dried storage has some effects on biochemical properties (Zornoza, Mataix-Solera, Guerrero, Arcenegui, & Mataix-Beneyto, 2009), we assumed these would be consistent across all treatments. The incubations were carried out at ~23 °C with a constant water content of 60% MWHC. The MWHC for each soil sample was calculated as the difference in weight between a saturated soil that was allowed to drain for 6 h and the weight after the soil was oven dried for 48 h at 105 °C (Haney & Haney, 2010; McDaniel & Grandy, 2016). To begin the incubation, ~5 g of soil, in 50-ml centrifuge tubes, was wet to 60% MWHC and incubated in the dark for 107 d. Soil water content throughout the incubation period was maintained by weight by adding deionized water to achieve 60% MWHC.

During the 107-d incubation, CO<sub>2</sub> production was measured using tunable-diode laser absorption spectroscopy (TGA 200A, Campbell Scientific) by injection of discrete gas samples into a carrier gas of CO<sub>2</sub>-free air (Hall, Huang, & Hammel, 2017). To capture the temporal changes in CO<sub>2</sub> production over the 107-d incubation, 18 measurements were taken more frequently at the beginning of the incubation when respiration rates were very high and less frequently toward the end of the incubation experiment as the respiration rate decreased. Each of the 18 CO<sub>2</sub> measurement events began by flushing tubes with ambient atmosphere, capping the tubes, and extracting a time-zero gas sample from the headspace via syringe and injecting it into the analyzer, followed by analysis of a second headspace sample between 5 h up to 3 d later. The time intervals between samples were dependent on the rate of CO<sub>2</sub> production, which decreased over the course of the experiment. Production of CO<sub>2</sub> was calculated as the difference in concentrations between the two time points divided by time. These production measurements occurred hourly or daily at the beginning of the experiment when CO<sub>2</sub> production was greatest and later occurred at weekly, biweekly, or monthly intervals toward the end of the incubation. Cumulative PMC was calculated by linear interpolation of CO<sub>2</sub> production between all 18 measurements.

Potential net N mineralization rates were determined from the same subsamples used for PMC incubations. Net N mineralized over the 107-d incubation experiment was calculated as the difference between inorganic N produced after 107 d of incubation and initial inorganic N extracted on separate soils before incubation. Inorganic N content of the soil was measured (5 g soil/25 ml KCl, mass/volume) using a 2 M KCl extraction method (Mulvaney, 1996). The extractant was analyzed colorimetrically on a Synerg HTX Multi-Mode Microplate Reader (BioTek Instruments, Inc.) for NH<sub>4</sub><sup>+</sup>-N using the salicylate and ammonia cyanurate reagent packets (Hach Company) at 595 nm and for NO<sub>3</sub><sup>-</sup>-N using the single-reagent method [vanadium III, sulfanilamide and N-(1-naphthyl)-ethylenediamine dihydrochloride] at 540 nm (Doane & Horwath, 2003).

## 2.4 | Data analysis

A normalized soil health recovery score (SHRS) was calculated to determine which topographic position exhibited the greatest improvement in soil health under CRP, controlling for inherent differences in topographic position. First, we calculated mean values for each soil health indicator for cropland ( $\bar{x}_{\text{crop}}$ ) and native grassland ( $\bar{x}_{\text{NG}}$ ) within each topographic position. Then, the relative recovery of each soil health parameter, or SHRS, was calculated using either of two equations. Equation 1 was used for soil health indicators where higher values equate to improved soil health (e.g., MWHC, SOC,

PMC, and PMN); Equation 2 was used when a lower value constitutes improved soil health (e.g., BD and inorganic N).

$$\text{SHRS}_i = \frac{x_{\text{CRP}} - \bar{x}_{\text{crop}}}{\bar{x}_{\text{NG}} - \bar{x}_{\text{crop}}} \quad (1)$$

or

$$\text{SHRS}_i = \frac{\bar{x}_{\text{crop}} - x_{\text{CRP}}}{\bar{x}_{\text{crop}} - \bar{x}_{\text{NG}}} \quad (2)$$

where SHRS is a normalized soil health score based on an individual soil health indicator value from the recovered grassland at each topographic position (i.e., CRP,  $x_{\text{CRP}}$ ) minus  $\bar{x}_{\text{crop}}$  divided by the difference in means ( $\bar{x}_{\text{NG}} - \bar{x}_{\text{crop}}$ ). The closer the SHRS value is to 1, the healthier the soil and the more similar it is to native grassland (our soil health benchmark or goal), whereas values close to zero are more like cropland soil.

To add context to our findings, we synthesized previous research on grassland restoration and soil C in North America. We focused on soil C for this synthesis due to the importance of both SOC and active C (whether measured as MBC or PMC) pools to soil health and ecosystem services. We used Web of Science (<http://apps.webofknowledge.com>) and Google Scholar (<https://scholar.google.com>) for our literature search and reviewed the literature with the criterion that manuscripts reported changes in soil C resulting from land conversion from cropland to reestablished grasslands in North America. The keywords and terms used for the online literature searching were “reestablished grassland OR perennial grassland OR conversion from agriculture to grassland” AND “Conservation Reserve Program” AND “soil organic matter OR soil organic carbon OR soil carbon OR microbial biomass OR potentially mineralizable carbon” AND “chronosequence OR prairie restoration chronosequence OR restored grassland chronosequence.” We then filtered studies based on a few criteria. First, we focused on studies that included a cropland and reestablished grassland treatment; native grassland was optional. Studies needed to also include the years-since-reestablishment and be based in North America. This resulted in 13 studies published in peer-reviewed journals from 1994 to 2019. Data were extracted from tables directly or from figures using data digitizer (GetData, <http://getdata-graph-digitizer.com/>). We collected average values from three treatments within each study: cropland soil, maximum reestablished grassland soil, and native grassland soil (if the study had native grassland). If native grassland data were available, we could calculate SHRS. We used the same SHRS framework to normalize both SOC and active C recovery across all 13 studies.

We checked the data for Shapiro–Wilk’s normality test and Barlett’s heterogeneity of variances test with the R statistical package (R Core Team, 2014). Normal distribution plots of data and heterogeneity of variances indicated unequal

**TABLE 1** Analysis of variance with years-since-reestablishment and topographic position as main factors

Parameters <sup>a</sup>	Factor <sup>b</sup>	Regression coefficient	P value <sup>†</sup>
BD	Y	−0.002	.1709
	TP	<b>−0.05</b>	<b>&lt;.001***</b>
	Y × TP	0.001	.1859
MWHC	Y	<b>0.004</b>	<b>.0036**</b>
	TP	<b>0.04</b>	<b>.0002***</b>
	Y × TP	<b>−0.002</b>	<b>.0003***</b>
SOC	Y	0.18	.1892
	TP	<b>4.83</b>	<b>&lt;.001***</b>
	Y × TP	<b>−0.09</b>	<b>.0586</b>
PMC	Y	<b>5.47</b>	<b>.0146*</b>
	TP	<b>39.2</b>	<b>.0019**</b>
	Y × TP	−0.11	.8846
Extractable inorganic N (NH <sub>4</sub> <sup>+</sup> −N + NO <sub>3</sub> <sup>−</sup> −N)	Y	−0.03	.6478
	TP	<b>2.33</b>	<b>&lt;.001***</b>
	Y × TP	−0.02	.4121
PMN	Y	0.50	.4649
	TP	<b>12.97</b>	<b>.0020**</b>
	Y × TP	−0.16	.5298

Note. Significant values are bold.

<sup>a</sup>BD, bulk density; MWHC, maximum water holding capacity; PMC, potentially mineralizable carbon; PMN, potentially mineralizable nitrogen; SOC, soil organic carbon.

<sup>b</sup>TP, topographic position; Y, years-since-reestablishment.

\*Significant at the .05 probability level.

\*\*Significant at the .01 probability level.

\*\*\*Significant at the .001 probability level.

variances. Assuming unequal variances, the main effects of time, topographic positions (shoulder-, back-, foot-, and toe-slopes) and their interaction was tested on all six response variables using mixed effects models with the R package “nlme” (Pinheiro, Bates, DebRoy, & Sarkar, 2018). Cropland was given the value of 0 yr, and reestablished grasslands ranged from 2 to 40 yr. Because the age of native grassland could not be determined, they were not included in time-dependent models of years-since-reestablishment. We used site as a random effect to account for spatial correlations among samples within a given site. Topographic position was modeled as a continuous, numerical predictor (1–4; arbitrary units) with increasing values from shoulder slope to toe slope. Similarly, time was modeled as a continuous numerical predictor (0–40 yr). The regression coefficients from the mixed effects models indicate magnitude and direction of the treatment effects.

Because we observed a significant main effect of CRP age on MWHC and PMC (Table 1), the effect of treatments (e.g., 0 yr as croplands, 2–40 yr in CRP, and native grassland) on those response variables were tested using a one-way

ANOVA. Tukey’s HSD test was used to determine the significant differences ( $p < .05$ ) among the treatments (i.e., years-since-reestablishment as a categorical variable). The test compared all possible pairs of means (e.g., native grassland to years-since-reestablishment and cropland and all CRP years to cropland). Coefficient of variation across the chronosequence, within years in CRP, was calculated for all response variables. Linear regression analysis was also done to isolate normalized SHRS and response variable changes by topographic positions using Sigmaplot 12.5 (Systat Software Inc.). The significance for all analyses was set at  $\alpha = .05$ .

### 3 | RESULTS

#### 3.1 | Soil bulk density and maximum water holding capacity

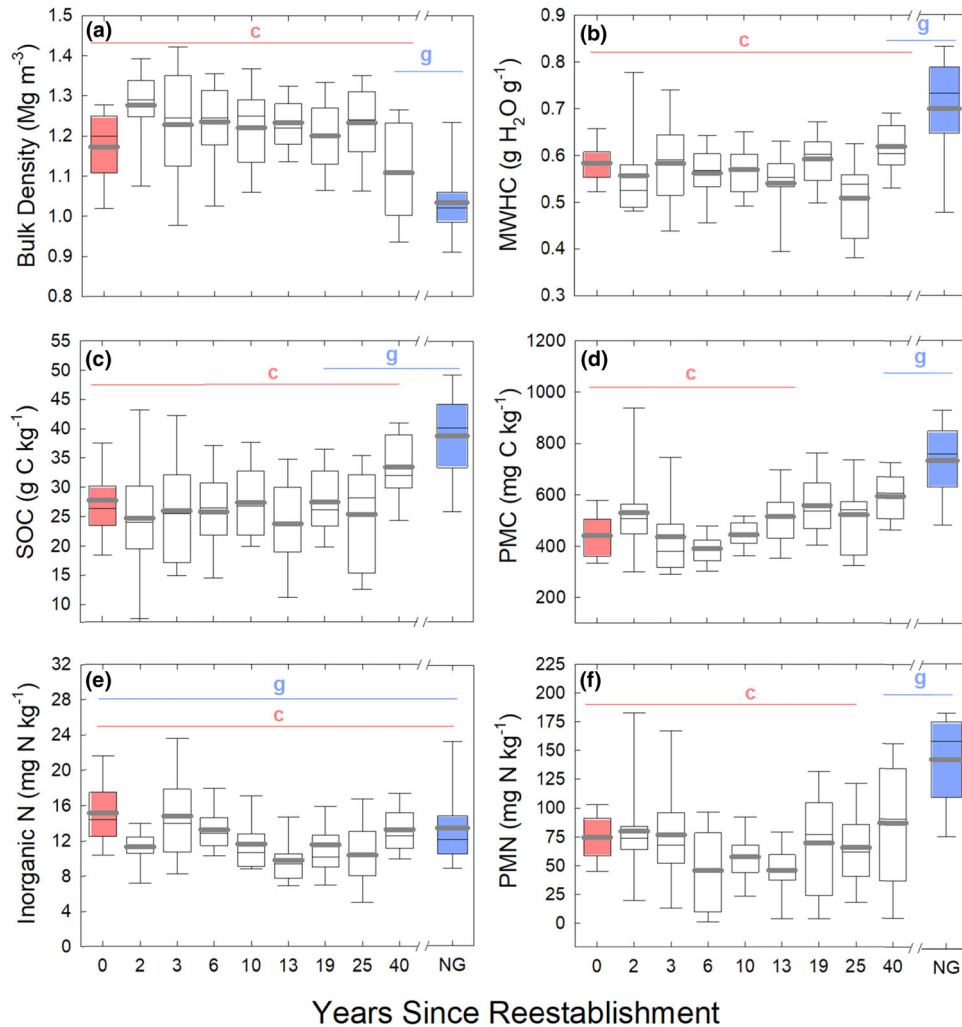
Years-since-reestablishment had no significant main effect on BD (Table 1). Reestablished grasslands increased BD in the first 2–20 yr of CRP to 1.23 Mg m<sup>−3</sup> compared with croplands at 1.15 Mg m<sup>−3</sup> (Figure 2a). However, after 3 yr, BD began slowly declining in reestablished CRP grasslands. Native grassland and 40 years-since-reestablishment soils had 8.6 and 4%, respectively, lower BD than the cropland soils (although nonsignificant). Overall, BD from our study soils did not seem very sensitive to CRP (Table 1). When isolating BD changes by topographic position, however, only foot slope position showed a significant decrease in BD with raw data compared with all other positions. However, using the normalized SHRS, only shoulder slope soils showed a significant linear recovery or a decrease in BD and reached 68% of native grassland by 40 yr ( $p = .036$ ) (Figure 3a).

The mixed effects models showed that both interactions and main effects on MWHC were significant with years-since-reestablishment and topographic position (Table 1). Across all topographic positions, the estimated time to reach native grassland MWHC at the increase of 0.004 g H<sub>2</sub>O g<sup>−1</sup> dry soil yr<sup>−1</sup> would be 30 yr (Table 1). Mean MWHC was greatest in native grassland soils (0.70 ± 0.02 g H<sub>2</sub>O g<sup>−1</sup> soil) and was 13–37% greater than the cropland and nearly all CRP soils (Figure 2b). Despite the significant interaction between years-since-reestablishment and topographic position (Table 1), there was no significant change in MWHC when the changes were isolated by topographic positions or when normalized using SHRS (Table 2; Figure 3b).

#### 3.2 | Soil organic carbon and potentially mineralizable carbon

Total SOC concentration did not show a significant main effect of years-since-reestablishment (Table 1). Mean SOC concentration of our benchmark soil (i.e., native grassland; 39 ± 1.7 g SOC kg<sup>−1</sup>) was 38% and 19–62% greater than the soils

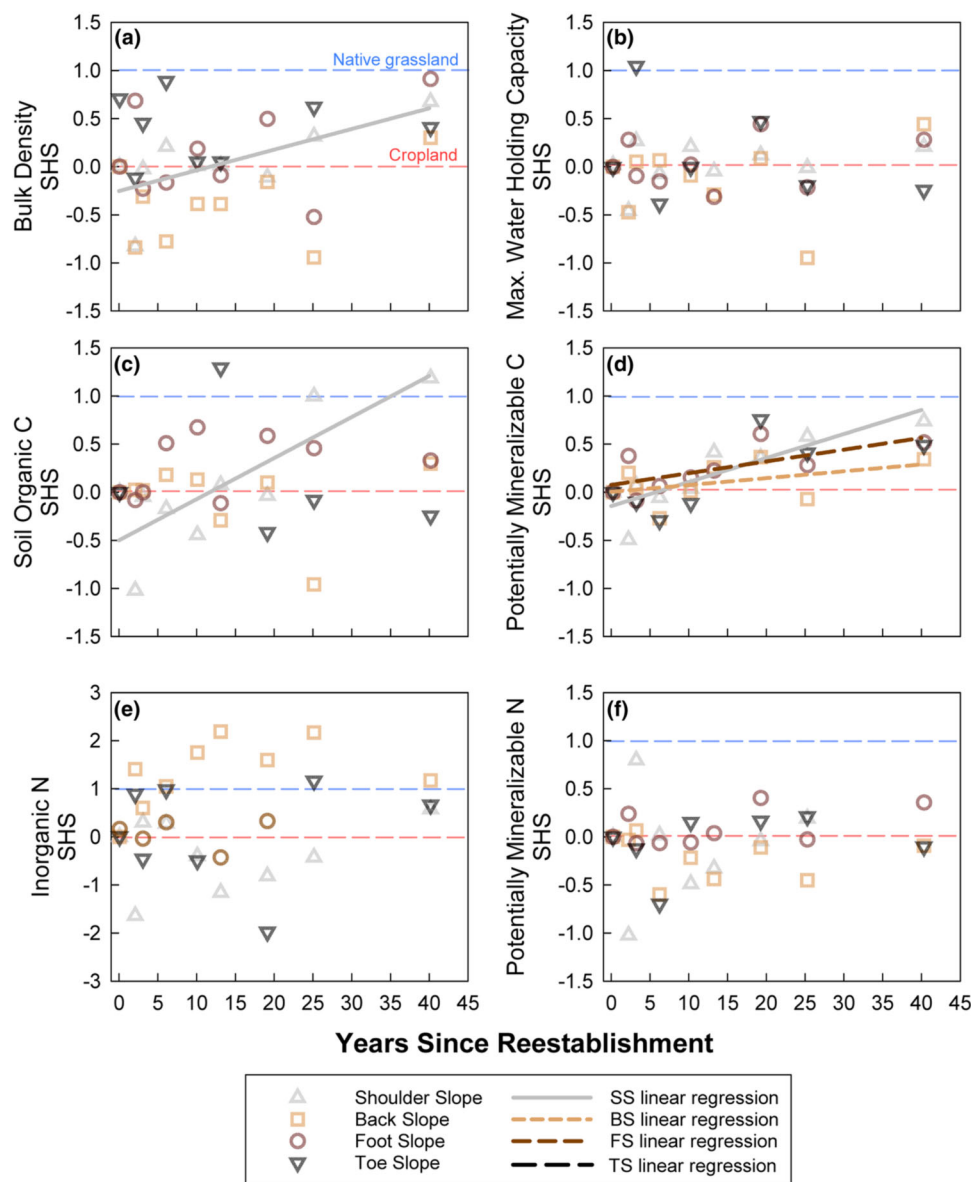




**FIGURE 2** Box plots showing (a) bulk density, (b) maximum water holding capacity (MWHC), (c) soil organic C (SOC) stock, (d) potentially mineralizable C (PMC), (e) extractable inorganic nitrogen ( $\text{NH}_4^+ - \text{N} + \text{NO}_3^- - \text{N}$ ), and (f) potentially mineralizable nitrogen (PMN) through years-since-reestablishment (0, cropland; NG, native grassland). Box plots show 10th, 25th, median, 75th, 90th percentiles. Means are shown as gray thick lines. Sample numbers in each box range from 12 (2 yr) to 42 (cropland). Lowercase c and g indicate no difference between cropland or native grassland mean, respectively

from cropland ( $28.4 \pm 1.3 \text{ g SOC kg}^{-1}$ ) and nearly all CRP soils ( $24\text{--}33 \text{ g SOC kg}^{-1}$ ), respectively (Figure 2c). Although SOC concentration did not show a significant relationship with years-since-reestablishment, we found a mean annual increase of  $0.18 \text{ g SOC kg}^{-1}$  of soil. At this rate, the conversion of cropland to CRP grasslands may take  $>51 \text{ yr}$  to recover to native grassland level. When the SOC changes by topographic positions were isolated, mean SOC concentration was highest in the toe slope position and was  $\sim 50\%$  greater than the shoulder slope position ( $25.1 \pm 1.1 \text{ g SOC kg}^{-1}$ ) (Table 2). However, compared with all other positions, only shoulder slope position showed a significant SOC increase with years-since-reestablishment and eventually exceeded native grassland levels by 19% by 40 yr when the changes were normalized using SHRS (Table 2; Figure 3c).

The PMC values ranged from 147 to  $1092 \text{ mg C kg}^{-1}$  (CV, 13–43%) along the CRP chronosequence (Figure 2d). The mean PMC content was highest in native grassland ( $733 \pm 35 \text{ mg kg}^{-1}$ ), which was 66% and 23–87% greater than the cropland ( $442 \pm 15 \text{ mg kg}^{-1}$ ) and the entire CRP chronosequence ( $391\text{--}594 \text{ mg kg}^{-1}$ ), respectively (Figure 2d). Although significant increases were observed at 19–40 yr in CRP compared with cropland, they only reached 71–81% of native grassland PMC (Figure 2d) and would require  $>53 \text{ yr}$  for a full recovery of cropland PMC to native grassland levels ( $\sim 5 \text{ mg PMC kg dry soil}^{-1} \text{ yr}^{-1}$ ) (Table 1). When the PMC changes with restoration were isolated by topographic positions, PMC showed strong linear trends at all topographic positions except toe slope (Table 2; Figure 3d). Soil health recovery was most rapid in the shoulder slope



**FIGURE 3** Normalized soil health recovery score (SHRS) for (a) bulk density, (b) maximum water holding capacity, (c) soil organic C stock, (d) potentially mineralizable C, (e) extractable inorganic nitrogen ( $\text{NH}_4^+ - \text{N}$  +  $\text{NO}_3^- - \text{N}$ ), and (f) potentially mineralizable N through years-since-reestablishment and by topographic positions. Native grassland (SHRS = 1) and cropland (SHRS = 0) soils are shown as horizontal, dashed blue and red lines, respectively. BS, back slope; FS, foot slope; SS, shoulder slope; TS, toe slope

soils and reached 74% of native grassland by 40 yr ( $p = .003$ ) (Figure 3d).

### 3.3 | Extractable inorganic nitrogen and potentially mineralizable nitrogen

Mean extractable inorganic N was highest in the cropland soil ( $15.1 \text{ mg N kg}^{-1}$ ), likely due to fertilizer or manure N inputs and tillage during agricultural management (Karlen et al., 1999). Extractable inorganic N ranged from 4.1 to  $34.2 \text{ mg N kg}^{-1}$  across the entire CRP chronosequence (Figure 2e).

Although CRP duration did not show a significant effect on extractable inorganic N content, we found an annual decrease of  $0.03 \text{ mg kg}^{-1}$  of soil for inorganic N pools (Table 1). Only back slope soils showed a significant decrease in inorganic N (Table 2), but, overall, extractable inorganic N from our study soils does not appear very sensitive to CRP, even when using normalized SHRS (Figure 3e).

In our study, PMN varied between 1 and  $206 \text{ mg N kg}^{-1}$  (CV, 43–73%) across the entire chronosequence of CRP (Figure 2f). Mean PMN content was highest in native grassland soil ( $143 \pm 9 \text{ mg kg}^{-1}$ ), which was 92% greater than the cropland soil and 64–212% greater than the entire

**TABLE 2** Linear regression parameters for years-since-reestablishment at each topographic position

Topographic position	df	Slope	P value	Mean $\pm$ SEM	Range
Bulk density ( $\text{Mg m}^{-3}$ )					
Shoulder slope	37	−0.003	.1103	$1.23 \pm 0.02$	0.95–1.45
Back slope	127	−0.001	.3886	$1.22 \pm 0.01$	0.91–1.45
Foot slope	29	<b>−0.005</b>	<b>.0036**</b>	$1.18 \pm 0.02$	0.96–1.35
Toe slope	16	0.002	.4392	$1.09 \pm 0.03$	0.79–1.28
Maximum water holding capacity ( $\text{g H}_2\text{O g}^{-1}$ soil)					
Shoulder slope	37	0.0003	.6708	$0.57 \pm 0.01$	0.43–0.68
Back slope	130	0.0003	.6331	$0.57 \pm 0.01$	0.35–0.80
Foot slope	30	0.002	.2012	$0.57 \pm 0.02$	0.35–0.74
Toe slope	15	−0.0024	.1779	$0.66 \pm 0.02$	0.39–0.84
Soil organic C ( $\text{g kg}^{-1}$ )					
Shoulder slope	38	<b>0.24</b>	<b>.0173*</b>	$25.1 \pm 1.1$	13.9–44.8
Back slope	128	0.002	.9696	$26.2 \pm 0.6$	4.2–49.5
Foot slope	30	0.25	.0148	$30.1 \pm 1.1$	18.2–41.8
Toe slope	17	−0.11	.5236	$37.7 \pm 2.4$	19.2–58.4
Potentially mineralizable C ( $\text{mg kg}^{-1}$ )					
Shoulder slope	37	<b>7.42</b>	<b>&lt;.0001***</b>	$479 \pm 21$	240–743
Back slope	130	<b>2.43</b>	<b>.0173*</b>	$474 \pm 10$	257–1002
Foot slope	30	<b>8.66</b>	<b>&lt;.0001***</b>	$463 \pm 26$	147–793
Toe slope	15	3.63	.2477	$627 \pm 43$	455–1092
Extractable inorganic N ( $\text{NH}_4^+ + \text{NO}_3^-$ -N; $\text{mg kg}^{-1}$ )					
Shoulder slope	37	−0.03	.5604	$11.6 \pm 0.6$	5.9–20.8
Back slope	130	<b>−0.10</b>	<b>.0011**</b>	$11.7 \pm 0.3$	4.2–24.9
Foot slope	30	−0.07	.4251	$14.2 \pm 1.0$	6.7–29.3
Toe slope	15	−0.11	.3783	$19.1 \pm 1.6$	11.0–34.2
Potentially mineralizable N ( $\text{mg kg}^{-1}$ )					
Shoulder slope	37	<b>1.18</b>	<b>.0470*</b>	$62 \pm 6$	7–158
Back slope	130	<b>−0.73</b>	<b>.0222*</b>	$60 \pm 3$	1–175
Foot slope	30	<b>1.36</b>	<b>.0118*</b>	$65 \pm 6$	6–146
Toe slope	15	−0.14	.8149	$110 \pm 8$	52–206

Note. Significant values are bold.

\*Significant at the .05 probability level.

\*\*Significant at the .01 probability level.

\*\*\*Significant at the .001 probability level.

CRP chronosequence. It was notable that PMN showed much more variability than other soil health measures and showed little progress toward reaching native grassland levels. Similar to PMC, PMN showed a strong linear trend with years-since-reestablishment at all topographic positions except for toe slope (Table 2). However, the SHRS was greatest in foot slope soils but only reached 36% of native grassland soils by 40 yr (Figure 3f).

## 4 | DISCUSSION

We used a unique chronotoposequence to examine the effect of grassland reestablishment on soil health and to evaluate

how topographic position alters this effect. By controlling for topographic position, we could better evaluate changes in soil health (Figure 3). Our findings are in line with other studies showing that ignoring topographic position can mask management effects on soils and soil processes (Kravchenko et al., 2005, 2006; Quigley et al., 2018). Moreover, our study clearly illustrates that topographic position can even mask changes from dramatic management practices, such as converting cropland to CRP grasslands. By extension, changes in soil health from other less dramatic soil conservation practices, like including winter cover crops or reducing tillage, could be masked by topographic position to an even greater extent within the U.S. Midwest Corn Belt.

#### 4.1 | Can reestablishing Conservation Reserve Program grasslands also restore soil health, and over what time scales?

For these questions, we were interested in whether the effect of years-since-reestablishment on soils would be strong and consistent enough to emerge as a significant main effect. Perhaps we should not expect reestablished grasslands to fully recover to native grassland levels over decadal scales because the loss in SOM in this region has been going on for at least a century. Disturbances due to cultivation or drainage alteration may hinder the ability of soil health to fully recover, with postcultivated soils reaching a new ecosystem steady state (Scheffer, Carpenter, Foley, Folke, & Walker, 2001; Schimel, Coleman, & Horton, 1985). Even soils under grassland cover for 19–40 yr showed little to no significant difference from cropland soils (Figure 2a, c), although some indicators were approaching native grassland soils regardless of topographic position (Figure 2b, d; Table 1). Therefore, the answer to the question as to whether reestablishing grasslands restores soil health is a conditional “yes.”

#### 4.2 | Soil bulk density and maximum water holding capacity

Bulk density reflects soil compaction and can be a proxy for other soil physical properties, such as porosity, root penetration resistance, and water holding capacity (Karlen et al., 1997; Murphy et al., 2004). We hypothesized that BD would decrease with time in CRP among the sites, and although there was a decreasing trend with grassland restoration age, BD was not significantly different from cropland soils. Ceasing tillage during initial conversion to CRP has been shown to temporarily increase BD because tillage can disrupt aggregates and soil structure, temporarily creating greater pore space and lowering BD over the short-term. However, over the long term, cultivation increases BD (Murphy et al., 2004). Therefore, it is common to find BD increases after ceasing tillage in no-till (Bhattacharyya, Prakash, Kundu, & Gupta, 2006; Cambardella & Elliott, 1993) and newly established grassland studies (Baer et al., 2002; Matamala et al., 2008; McKee et al., 2019; Murphy et al., 2004). In our study, however, after 3 yr BD appeared to slowly decline toward native grassland values (Figure 2a), suggesting improved soil structure and porosity with grassland age (Matamala et al., 2008; Murphy et al., 2004; Rosenzweig et al., 2016; Scott et al., 2017). Over longer periods than we studied (>40 yr), increased root growth and higher SOM inputs (e.g., root residues, rhizodeposits, and surface litter) can enhance macroaggregate development in finer-textured soils and likely contributed to ~4% decreases in BD compared with cropped soil in other studies (Cambardella & Elliott, 1993; Jastrow,

1987). Other studies (e.g., Rosenzweig et al. [2016]) found relatively rapid reduction in BD (21%) across a 35-yr chronosequence of prairie restoration in eastern Kansas, with an estimated time to reach native grassland of 17 yr. Declining BD, as a function of years-since-reestablishment, may affect many other soil processes, such as improving infiltration rates, easing root penetration, increasing soil aeration, and reducing surface runoff and erosion (Foster, Young, Ronkens, & Onstad, 1985; Murphy et al., 2004).

Maximum water holding capacity is an indicator of soil water storage and is often related to plant available water (Hudson, 1994; Huntington, 2007). The MWHC was one of two variables that showed significant main effects of years-since-reestablishment, but it took ~30 yr to recover. This slow but steady increase in MWHC could improve local soil hydrologic functions (Ouyang et al., 2016). Further, improved soil water retention due to grassland restoration could help flood mitigation with changing climate and reduce sediment and nutrient transport or increase plant available water in drought years where grasses are used for grazing or biomass purposes (Li et al., 2017b). Not only was this one of the more sensitive variables to years-since-reestablishment (Table 1), but it also was one of the most inexpensive and easiest soil health indicators to measure using the “filter-paper method” to get gravimetric water held at field capacity (Fawcett & Collins-George, 1967; Haney & Haney, 2010). These three factors (management sensitivity, inexpensiveness, and ease to measure) are critical for adoption of any soil health indicator and make MWHC more likely to play a role in guiding land managers wanting to adopt soil health-promoting practices (Doran & Zeiss, 2000).

#### 4.3 | Soil organic carbon and potentially mineralizable carbon

Cultivation depletes SOC in native grasslands and is caused by multiple cropping practices, including tillage (Blanco-Canqui & Lal, 2009; Gebhart et al., 1994; Reicosky, Dugas, & Torbert, 1997); aeration through tile drainage, which is common for this region (Arabi et al., 2006; Du et al., 2005); reduction in plant C inputs to soil (Anderson & Coleman, 1985; McConnell & Quinn, 1988; Richter et al., 1990); and even liming of soils (Chan & Heenan, 1999; Paradelo, Virto, & Chenu, 2015; Wang et al., 2016). Several studies found increased SOC with similar chronosequence approaches (Baer et al., 2002; Baer, Meyer, Bach, Klopff, & Six, 2010; Matamala et al., 2008; McLauchlan, Hobbie, & Post, 2006). In addition, many studies highlighted that the rate of SOC change is nonlinear, usually with a faster recovery at the early stages of restoration (i.e., <6 yr) compared with late stages of restoration (i.e., >11 yr) (Hernández, Esch, Alster, McKone, & Camill, 2013; Kucharik, 2007). In our study,



without considering topographic position, there was no significant increase in SOC even up to 40 yr of grassland restoration in this region of the midwestern United States. Others have found a similar lack of change in SOC with grassland restoration and estimated that SOC levels in reestablished grasslands may take centuries (150 to >400 yr) to match that of native grasslands (Baer et al., 2010; Hernández et al., 2013; Matala et al., 2008; O'Brien & Jastrow, 2013; Rosenzweig et al., 2016). Expecting SOC to recover within 40 yr may be unrealistic, especially for midwestern U.S. soils that are already rich in SOM.

Potentially mineralizable C is an integrated measure of microbial biomass (Anderson & Domsch, 1978), activity (Wang, Dalal, Moody, & Smith, 2003), and availability of labile or active C (Franzluebbers, Haney, Honeycutt, Schomberg, & Hons, 2000; Wang et al., 2003). Unlike SOC, PMC recovered within 13–19 yr of grassland reestablishment. Although significant increases were observed over a relatively short time in our study, it would still take >53 yr for a full recovery of cropland PMC to native grassland levels (Table 1; Figure 2d). In contrast, Rosenzweig et al. (2016) found that PMC in a 35-yr reestablished grassland even exceeded that of native grassland within three decades. Other studies have also demonstrated more rapid recovery in labile C pools than total SOC pools (Baer et al., 2002, 2010; McLauchlan, 2006; Robles & Burke, 1998), suggesting this may be a more general phenomenon (Table 3). The greater sensitivity of PMC to management relative to SOC is likely due to greater root biomass, increased active C inputs, and associated increases in microbial biomass and activity (Baer & Blair, 2008; Maher, Asbjornsen, Kolka, Cambardella, & Raich, 2010; Rosenzweig et al., 2016). Other studies have suggested that PMC is as an early indicator of soil C accrual (Baer et al., 2010; McLauchlan, 2006; Sprunger & Robertson, 2018) and N-supplying power (Franzluebbers, 2018; Franzluebbers & Pershing, 2018), highlighting its importance as a soil health indicator.

#### 4.4 | Extractable inorganic nitrogen and potentially mineralizable nitrogen

Extractable inorganic N represents a snapshot of plant-available N but also likely reflects N that is easily leached through the soil profile (especially  $\text{NO}_3^-$ -N). High potential for N retention (via plant uptake and microbial immobilization) within reestablished grasslands could help reduce N export compared with cropland by reducing leaching and nitrification/denitrification losses (Dodds et al., 1996; Smith et al., 2013). We hypothesized that inorganic N would decrease with time-since-reestablishment, suggesting less residual inorganic N and fewer N losses (i.e., tighter N cycling) from grasslands as a function of years-since-

reestablishment (Dell & Rice, 2005; Rosenzweig et al., 2016). Our findings were generally consistent with our hypothesis, although they were not significant (Figure 2e; Table 1). Given that we sampled soils in July and August, most fertilizer N applied to cropped soils would have been assimilated by crops or lost via leaching or gases. Therefore, our data only reflect a snapshot of plant-available N (or residual N) remaining after fertilization, at peak plant biomass. However, there is still a great deal of uncertainty when using extractable inorganic N to assess growing season N dynamics and what that means for soil health. Overall, extractable inorganic N was not very sensitive to grassland reestablishment in our study, but others have reported inorganic N levels lowering to native grassland levels after 7 yr (Rosenzweig et al., 2016).

Potentially mineralizable N, an important indicator of soil health, is a proxy of the in situ N-supplying power of the soil and is a measure of N liberated in the same labile soil organic matter as PMC (Mahal, Castellano, & Miguez, 2018). We hypothesized this would increase with years-since-reestablishment, and our findings were more or less consistent with this, albeit not significant. It was surprising that PMN was not as sensitive as PMC to grassland restoration even though they were measured during the same incubation. It was notable that PMN showed high variability (CV, 43–73%); this might reflect microbial N limitation or  $\text{NH}_4^+$  fixation by clay minerals in some of these soils (although not measured here). Further, PMN will take longer than PMC to recover under the CRP, and 40 yr is not adequate time for reestablished grasslands to fully recover. Even controlling for topographic position did not help resolve restoration effects (Tables 1 and 2). Thus, this constraint is difficult to overcome at the decadal scale, and future studies may wish to focus on gross N mineralization rates or to quantify fixed  $\text{NH}_4^+$  to more accurately assess changes in soil N dynamics.

#### 4.5 | What role does topographic position have on regulating effects of the Conservation Reserve Program on soil health?

Contradictory findings among our study and previous CRP chronosequence studies may be, in large part, due to high spatial variability in soil properties that are masking any treatment effect (Kravchenko et al., 2005, 2006; Quigley et al., 2018; Wickings, Grandy, & Kravchenko, 2016). Because topography is one of the five soil forming factors (Jenny, 1941), it seems critical to account for it in any assessment of grassland restoration and recovery of soil health (Zilverberga et al., 2018). To our knowledge, we are the first study to use a “chronotoposequence” approach. Because topographic position had a significant main effect on all of our soil health indicators (Table 1), using it as an interacting

**TABLE 3** Rates of soil carbon (C) accrual, average total soil organic carbon (SOC) and active soil C contents during grassland reestablishment after conversion from croplands in North America. If the required information was missing, it was marked as not detected (ND)

Study reference	Location in USA or Canada	Max. years in restor.	Soil depth cm	Clay content %	Mean annual temp. °C	Mean annual precip. mm	Total SOC			Active soil C <sup>a</sup>		
							Cropland	Max restored	Native grassland	Cropland	Max restored	Native grassland
							g m <sup>-2</sup>	g m <sup>-2</sup>	g m <sup>-2</sup>	g m <sup>-2</sup> yr <sup>-1</sup>	g m <sup>-2</sup>	g m <sup>-2</sup>
Gebhart et al. (1994)	TX, KS, NE	5	300	ND	ND	430–500	5920	6510	9080	110.0	ND	ND
Knops and Tilman (2000) <sup>b</sup>	MN	61	10	1–4	6	775	ND	1638	2713	19.7	ND	ND
Baer et al. (2002)	NE	12	10	27–42	11	755	2048	2415	3641	29.5	12.1 (23.7)	49.0 (53.2)
Mensah et al. (2003) <sup>c</sup>	Saskatchewan	12	15	ND	ND	ND	1146	1750	ND	70.0	ND	ND
McLauchlan et al. (2006)	MN	40	10	11–30	6	650	2491	4983	6263	62.0	57	111.4
Matamala et al. (2008)	IL	26	15	25–31	9.4	937	4714	5549	7527	43.0	62.7	140.3
Baer et al. (2010)	NE	19	10	15–39	9.8–11	685–757	1704	2045	3103	21.2	48 (22.2)	95 (42.9)
Guzman and Al-Kaisi (2010)	IA	14	15	27–29	ND	ND	2997	4043	5524	74.0	ND	ND
Hernández et al. (2013)	MN	16	20	ND	ND	ND	1328	1553	ND	22.5	ND	ND
Rosenzweig et al. (2016) <sup>d</sup>	KS	35	10	20–33	ND	834	1670	2584	3675	26.2	14.4 (12)	53 (36.3)
Scott et al. (2017)	KS	35	10	ND	12.7	835	1604	2616	3465	23.5	18.9	70.6
Bugeja and Castellano (2018)	IA	21	15	20–43	ND	750	2455	3697	5777	59.1	ND	ND
McKee et al. (2019)	AR	38	10	8–17	13.7	1150	1452	ND	2735	–11.3	ND	ND
	Regional range or mean ± SE	5–61	10–300	1–43	6–13.7	430–1150	2552 ± 451	3141 ± 464	4864 ± 641	42.3 ± 8.8	35.5 ± 9.4 (19.3 ± 3.7)	86.6 ± 14.6 (44.1 ± 4.9)
Our study	IA, MN	40	15	17–34	6–9	625–850	4895	5397	5973	21	(77.1)	(97.0)

<sup>a</sup>Active soil C data represent both microbial biomass C (MBC) and potentially mineralizable C (PMC); PMC values are in parentheses.  
<sup>b</sup>Knops and Tilman (2000): We converted soil C data from percent to area basis (g m<sup>-2</sup>) assuming bulk density of 1.3 g cm<sup>-3</sup> at 10 cm soil depth.  
<sup>c</sup>Mensah et al. (2003): We used the mean SOC values from 0- to 5-cm depth for both cropland and restored grassland treatments among five restoration sites.  
<sup>d</sup>Exponential equation.

variable with years-since-reestablishment did help to resolve some differences in soil health recovery. Using both a raw data approach (Table 2) and the normalized SHRS approach (Figure 3), we showed that recovery of soil health depended on the topographic position. Across all six soil health measures, the shoulder slope position was the most sensitive to grassland restoration, showing the greatest recovery at 40 yr (Figure 3a, c, d), supporting our hypothesis that higher topographic positions would have greater improvements in soil health from CRP. This is most likely because the shoulder slope soils were more affected and degraded by erosion during cultivation (Olson et al., 2016a, 2016b; Zilverberga et al., 2018).

In contrast to our study, a 12-yr grassland restoration study in Saskatchewan, Canada, found 88–169% increases in SOC stock from higher to lower topographic positions, respectively (Mensah et al., 2003). This, the authors suggested, was due to greater soil moisture both increasing plant biomass production and lowering rates of decomposition at lower landscape positions. Their contrasting finding may reflect differences in climate and soil type between our studies, even though they did not measure the interaction between years-since-reestablishment and topographic positions.

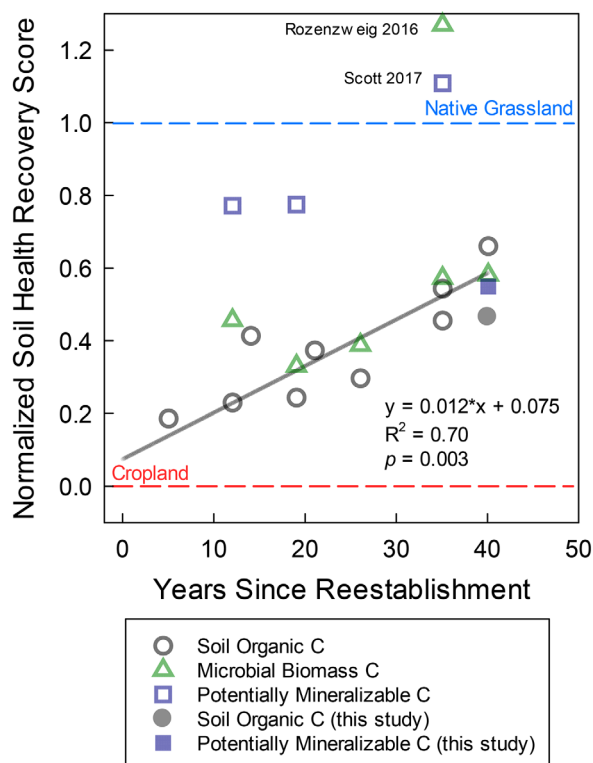
In contrast to our hypothesis, active C (measured as PMC) showed similar recoveries across most topographic positions, depending on how the data were analyzed (Table 2; Figure 3d). Assuming that PMC is an effective integrator of soil microbial metabolic potential, this evidence provides strong impetus for using PMC as a sensitive yet consistent index of soil health. In other words, PMC was sensitive enough to management (i.e., grassland restoration in this case) that a signal emerged through the noise caused by topographic position. Although topographic position is important in regulating all aspects of soil health (Table 1) and as an explanatory variable for basic soil science research, it is perhaps of less importance to land managers looking for feedback on their management impacts on soil health. In fact, trying to account for topographic variation may be a deterrence to land managers because collecting more soil samples by topographic position can be difficult and time-consuming (especially if there are no clear differences in topographic position). Not to mention, it could also add extra costs to analyzing a field's soil health. An ideal soil health indicator used to inform conservation practices should show similar changes to management regardless of inherent soil properties like topographic position. For this reason, we recommend PMC as the best general indicator of soil health recovery in our soils (Figure 3d), and PMC measured in short-term incubations is rapidly gaining popularity in the soil health research community (Franzluebbers, 2018; Wade, Horwath, & Burger, 2016; Wade et al., 2018).

## 4.6 | Our findings in the context of North American grassland reestablishment studies

Native grassland SOC, MBC, and PMC stocks were 53–151%, 164–668%, and 120–183% greater than cropland soils, respectively, across all North American studies (Table 3). Even though the total SOC stock in our study did not show a significant difference between reestablished grassland and cropland soil or a significant relationship with years-since-reestablishment (without controlling for topographic position), we found an overall mean increase of  $21.1 \text{ g SOC m}^{-2} \text{ yr}^{-1}$ . This compares with the range in accrual rates from  $-11.3 \text{ g SOC m}^{-2} \text{ yr}^{-1}$  in Arkansas soils (McKee et al., 2019) to  $110 \text{ g SOC m}^{-2} \text{ yr}^{-1}$  in a study across Texas, Kansas, and Nebraska (Gebhart et al., 1994). However, comparisons were confounded by differences in sampling depth. The North American regional average was  $42.3 \text{ g SOC m}^{-2} \text{ yr}^{-1}$ , not including our study.

Unlike SOC, recovery of active C (e.g., MBC and PMC) was much more variable across the selected North American studies that measured it (Table 3). Even rates of recovery between MBC and PMC were extremely variable, with MBC recovery rates ranging from  $1.2$  to  $5.7 \text{ g C m}^{-2} \text{ yr}^{-1}$  compared with PMC recovery of  $0.04$ – $0.69 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Nineteen years of grassland reestablishment increased PMC to 93% compared with cropland soils but to only 88% of native grassland PMC (Table 3) (Baer et al., 2010). One study showed similar PMC after 35 yr of restoration ( $36.3 \text{ g PMC m}^{-2}$ ) compared with native grassland soils ( $33.9 \text{ g PMC m}^{-2}$ ) (Rosenzweig et al., 2016).

Some interesting trends emerged when applying the normalized SHRS to synthesize the North American grassland chronosequence studies (Figure 4). First, a distinct linear trend in recovery of SOC emerges across the 14 studies (including ours): SOC recovers at about 1% per year under grassland reestablishment in North America. Second, measures of active C do not show a linear recovery trend, even when normalized for starting and endpoint (i.e., cropland and native grassland), which makes comparison among studies difficult. Third, two studies actually showed soil health recovery to levels greater than native grassland (Rosenzweig et al., 2016; Scott et al., 2017) (Figure 4); these levels were 11 and 27% greater PMC and MBC in reestablished grassland compared with the native grassland soils. Variation in SOC and active C recovery may depend on a variety of factors, such as initial soil C deficit, depth of measurement, topography (as demonstrated clearly in this study), parent material, climatic conditions (e.g., mean annual temperature, and precipitation), initial reestablishment methods, and management of reestablished grasslands (e.g., grazing, burning, and bailing). Among these factors, soil texture has received considerable attention as an important factor in regulating SOC and other soil health



**FIGURE 4** Normalized soil health recovery score (SHRS) from 13 studies in North America (open symbols) and this study (filled symbols). Native grassland (SHRS = 1) and cropland (SHRS = 0) soils are represented by horizontal, dashed blue, and red lines, respectively. Solid gray line represents regression line for soil organic C across the 10 studies (including ours)

indicators (Baer et al., 2010; Matamala et al., 2008; Mensah et al., 2003). In our study, texture did not play a major role in SOC changes, as found in some other studies (McLauchlan, 2006; McLauchlan et al., 2006). This can likely be attributed to the relatively narrow range of soil textures among our sites (Supplemental Figure S1).

According to USDA-FSA (2017), ~9.5 million ha of cropland is currently enrolled in CRP in the United States and may have the potential to sequester about  $4 \times 10^6$  Mg of C  $\text{yr}^{-1}$  at 15 cm soil depth at the mean increase of  $42.3 \text{ g SOC m}^{-2} \text{ yr}^{-1}$  (Table 3). However, these rates of C sequestration should not be expected to continue indefinitely because the soil C sink will gradually decline with time when soil C stocks approach saturation (Stewart et al., 2007). At the rate of  $4 \times 10^6$  Mg C  $\text{yr}^{-1}$ , 9.5 million ha currently enrolled in CRP could restore  $40 \times 10^6$  Mg C within one decade. Our estimated decadal C sequestration is greater than a previous estimated range of  $3\text{--}10 \times 10^6$  Mg C in CRP grasslands (Follet, 1993) but was within the estimated range of  $19\text{--}105 \times 10^6$  Mg C in North American grasslands restoration literature (Table 3). Because  $4 \times 10^6$  Mg  $\text{yr}^{-1}$  is a very small fraction of the C released from global annual anthropogenic C emis-

sions ( $10.7 \text{ Pg C yr}^{-1}$ ; Lal [2018]), the other enhanced soil ecosystem services from increasing SOC in North America are arguably a more important benefit than SOC sequestration per se.

## 5 | CONCLUSIONS

In the midwestern United States, reestablishment of CRP grasslands had minor effects on many soil health indicators when compared with croplands, even when established for up to 40 yr. Active C (measured as PMC) and MWHC, however, showed signs of improvement after 13–40 yr in CRP regardless of topographic position. Considering topography did increase our ability to detect soil health recovery from reestablishing grasslands, and we found the greatest soil health recovery in soils higher in the landscape. Out of six soil health indicators, PMC showed the most promise as a sensitive soil health indicator, with consistent increases with years-since-reestablishment at all but one topographic position. Thus, land managers, natural resource managers, and policy makers may want to consider using PMC as an indicator of any management practices' ability to improve soil health.

About 9.5 million ha of cropland currently enrolled in CRP may ultimately have the potential to improve soil health and sequester about  $4 \times 10^6$  Mg  $\text{yr}^{-1}$  of atmospheric C, but patience might be needed to observe soil changes, which may take decades rather than years. Thus, CRP has a small potential to mitigate climate change in addition to several other "climate-smart" agricultural practices (e.g., reduced tillage, perennial crops, and winter cover crops) (Lal, 2004a, 2004b; Paustian et al., 2016). There is a need for further research into how grassland species or functional types (e.g.,  $C_3$  vs  $C_4$  grasses) interact with topographic position to affect soil health, with an emphasis on studies of SOC changes at depths greater than 15 cm.

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## CONFLICT OF INTEREST

The authors declare no conflict of interest.


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## SUPPORTING INFORMATION

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